Nature Conservation 16: 19–37 (2016)
doi: 10.3897/natureconservation.16.9312
http://natureconservation.pensoft.net

RESEARCH ARTICLE

A peer-reviewed open-access journal
Nature Conservation
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Analysing fragmentation in vulnerable biodiversity hotspots in Tanzania from 1975 to 2012 using remote sensing and fragstats

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Academic editor: *M. Kleye* | Received 23 May 2016 | Accepted 30 October 2016 | Published 18 November 2016

http://zoobank.org/823FB494-A057-479B-9EFF-445F81F6E03A

Citation: Ojoyi MM, Odindi J, Mutanga O, Abdel-Rahman EM (2016) Analysing fragmentation in vulnerable biodiversity hotspots in Tanzania from 1975 to 2012 using remote sensing and fragstats. Nature Conservation 16: 19–37. doi: 10.3897/natureconservation.16.9312

Abstract

Habitat fragmentation is a threat to conservation of biodiversity hotspots in the Morogoro region, Tanzania. However, on-going research on fragmentation has not kept pace with temporal lapses and how individual species respond to habitat transformation and heterogeneity. This study sought to model spatial and temporal fragmentation patterns. Cloud free multi-temporal Landsat imagery with similar spectral resolution were acquired in the same season in 1975, 1995 and 2012. The images were used to characterize the biophysical landscape characteristics and a range of metrics used to quantify the magnitude of fragmentation. Patches and classes in the landscape were assessed using Fragstats, a spatial statistics program useful in computing landscape metrics. Results show that patch number was higher in dense forest and woodland than in less dense forest and grassland in 1975, 1995 and 2012 while the interspersion Juxtaposition Index (IJI) ranged between 0 (for clumped patches) and 100 (for grassland). In 1975 and 1995, the grassland habitat had the highest IJI while in 2012 less dense forest had the highest IJI. The Games-Howell test showed a significant fragmentation trend in less dense forests class ($p \leq 0.05$). Generally, the study indicates a high fragmentation pattern in the vulnerable tropical eastern arc mountain region of East Africa. This finding demonstrates the value of remotely sensed data in understanding the impact of anthropogenic processes on natural landscape transformation. Furthermore, the study provides a basis for informed conservation policy design and implementation in the region.

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Keywords
habitat, fragmentation, fragstats, remote sensing, Tanzania

Introduction

Habitat fragmentation, an indication of habitat transformation, degradation and loss is a great concern globally (Fahrig 2001, Fahrig 2003, Cushman et al. 2010, Forman and Godron 1986). It refers to habitat breakages or the degree of patchiness of a habitat, mainly as a result of anthropogenic impacts (Fahrig 2001, Fahrig 2003, McGarigal and Cushman 2002, Wiens 1995, Neel et al. 2004). Generally, habitat fragmentation interferes with the structural configuration of ecosystems and their ecological functioning (Abdullah and Nakagoshi 2007, Echeverria et al. 2006, Echeverria et al. 2008, Iida and Nakashizuka 1995, Forman and Godron 1986). Specifically, fragmentation reduces total habitat area, making species highly vulnerable to endemism and extinction (Yen et al. 2005, Murcia 1995, Aguilar et al. 2008, Yen et al. 2005). Hence fragmentation has long term impacts on species numbers (Aguilar et al. 2008) and species abundance (Fahrig 2003, Debinski and Holt 2001, Yen et al. 2005, Forman and Godron 1986) as it exposes natural ecosystems to external risks such as parasitism and dominance of invasive species (Wiens 1995).

Habitat fragmentation is an explicit challenge to conservation in the tropics (Vogelmann 1995). It is considered a major cause of species loss (Pelkey 2000, Adams et al. 2003, Bjørndalen 1992, Burgess et al. 2002, Burgess et al. 2001, Yen et al. 2005, Forman and Godron 1986). In Africa, approximately 310,000 hectares of forest is annually converted to agriculture, while 200,000 hectares is converted into woodlands, major causes of fragmentation (Achard 2002). Fragmentation acts synergistically with other factors like effects of solar radiation and open niches that lead to dominance of other invasive species. Consequently, native vegetation species are exposed to higher risks of extinction with a decline in the percentage area required for their survival (Rutledge 2003).

Ecosystems in Morogoro region, Tanzania contribute to the world's climate regulation through large carbon stores (Burgess et al. 2007, Swetnam et al. 2011). These forests are also characterized by high levels of endemism and many species are vulnerable to extinction (Swetnam et al. 2011, Brooks et al. 2006, Myers et al. 2000). Increased anthropogenic disturbances in particular pose significant threats to their long term conservation (Hall et al. 2009, Hall 2009, Newmark 1998). Between 1955 and 2000 for instance, forest cover declined from 300 km^2 to 220 km^2 (Burgess et al. 2007). Despite the area's global importance, few studies have been conducted with a focus on its spatial heterogeneity (Newmark 1998). Furthermore, mechanisms by which natural habitats respond to spatial heterogeneity across diverse fragmenting ecosystems remain largely unexplored (Swetnam et al. 2011, Yanda and Shishira 1999). Individual habitats may differ in their degree of response to fragmentation as the robustness of fragmentation may vary (Fahrig 2003, Neel et al. 2004, McGarigal 2006,

Echeverría 2007). For instance, due to differences in their structural complexity and biological processes, what could be termed as fragmentation in homogeneous landscapes may be interpreted differently in a heterogeneous landscape (Murcia 1995, Fischer and Lindenmayer 2007, Wiens 2000). In this study, we tested the spatial extent and magnitude of fragmentation in four vulnerable habitats subjected to fragmentation in the region. Remote sensing was applied due to its increasing popularity in quantifying spatio-temporal patterns in diverse landscapes (Ojoyi et al. 2016, Nagendra et al. 2004, Lung and Schaab 2006, Southworth et al. 2002, Fjeldså 1999). Specifically, the study pursued the following objectives: (1) to investigate multi-temporal magnitude of fragmentation in diverse habitats; (2) to quantify the intensity of habitat fragmentation in each of the habitats.

Study area

Similar to this case study, most rich biodiversity hotspots in Tanzania are geographically located in the Eastern Arc Mountains (Burgess et al. 2007, Myers et al. 2000, Hall et al. 2009, Hall 2009, Newmark 1998, Olson and Dinerstein 1998). In this study, we selected a section of Morogoro region dominated by four major habitat types (Figure 1). The choice of the study location was based on previous ecological studies (e.g. Burgess et al. 2002, Burgess et al. 2001, Hall 2009, Luoga et al. 2000, Yanda and Shishira 1999) that attributed species losses to fragmentation. The study area is characterized by sub-montane (with trees 30-50m tall), montane (with trees 15-30m tall) and upper montane (with trees 15-20m tall) forest at 1200-1500, 1500-2100 and >2100m asl, respectively. Generally, forest density and height varies with elevation and aspect, with dense canopy dominating lower altitude and elfin forests dominating ridges above 1900m asl (Lovett et al. 1996). Stunted grass patches are also common at high altitudes. According to Burgess et al. (2002), the potential of closed natural forest cover is about 500km^2, however, this has been reduced from 300km^2 in 1955 to 230 km^2 in 2001, with most decline recorded at 600-1600 m asl outside protected areas. The loss is mainly attributed to increasing population growth, estimated at about 2.5-3% per annum (Lovett 1996). The study area comprises four main habitats; woodland, dense forest, less dense forest and grassland. In this study, woodland is described as woody vegetation with scattered foliage cover (less than 30%) with mature stands of less than five meter tall while less dense forest consists of fields and patches with trees of more than six meters tall, with crown cover of less than 30%. The dominant tree species in the region include: *Bersama abyssinica*, *Cassipourea malosana*, *Cornus volkensii*, *Cussonia lukwangulensis*, *C. spicata*, *Dombeya torrida*, *Draceana afromontana*, *Garcinia volkensii*, and *Xymalos monospora*. Bamboo thickets form dense stands of *Sinarundinaria alpina* 12-15 m tall and 15 cm diameter (Bjørndalen 1992, Shirima et al. 2011, Lovett 1993). The grassland habitat consists of *Panicum lukwangulense* and *Andropogon thystinus* with scattered trees of *Agauria saliciflora*, *Adenocarpus mannii*, *Myrica salicifolia* and *Berberis* sp. thought to have replaced upper montane forest due to fires (Adams et al.

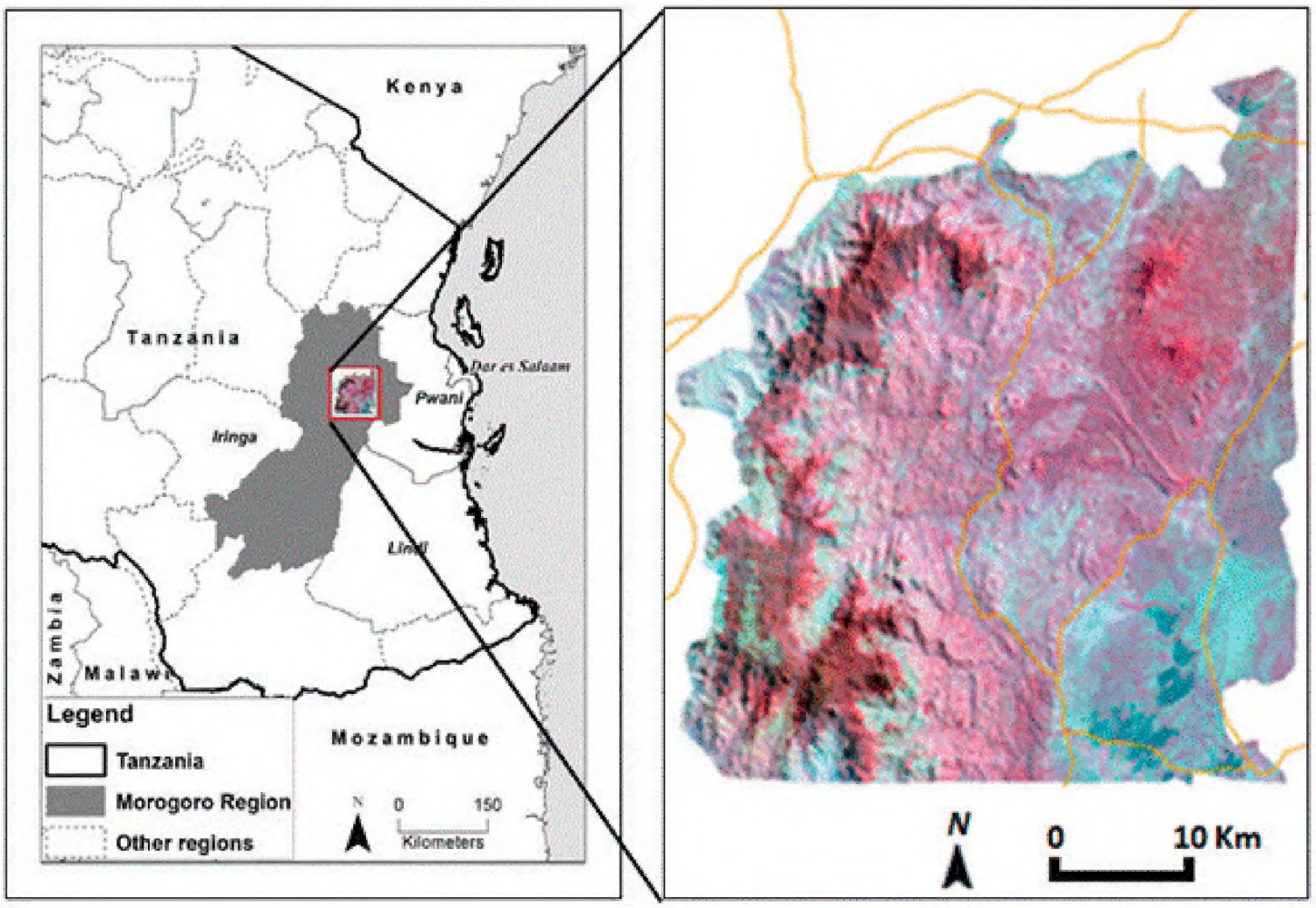

Figure 1. A 1975 Landsat composite overlaid on the study area.

2003, Bjørndalen 1992). Kitulanghalo forest is located between Morogoro and Dar es Salaam within Morogoro region. The woodland forms part of Miombo woodland which covers 90% of the total forested ecosystem (Mugasha et al. 2013, Munishi et al. 2010). It is dominated by *Brachystegia*, *Isoberlinia*, *Julbernardia*, *Pterocarpus angolensis*, *Afzelia quanzesis* and *Albizia species* (Chander 2009). The area falls within a semi-natural Miombo woodland which receives less than 1000 mm of rainfall per annum. Their proximity to Morogoro urban area increases their susceptibility to anthropogenic activities, altering their functioning and sustainable management (Mugasha et al. 2013).

Methodology

Image pre-processing

Landsat MSS (20/08/1975), Landsat TM (30/09/1995) and Landsat ETM+ (20/07/2012) imagery with better visualization (less than 15% cloud cover) from the Global Landcover Facility (http://www.landcover.org) were selected for the study. Datum was set to WGS 84 and referenced to Universal Transverse Mercator (UTM) zone 37 South. All images were orthorectified using ground validation points, Digital Elevation Model (DEM) and aerial photos as a reference. Landsat images were resampled to a common resolution pixel (30 × 30 m) using bilinear resampling to ensure consistency

in all image scenes. First order polynomial transformation was applied at image registration to correct for any shifts. It was deemed necessary to simulate atmospheric interactions between the sun and sensor pathways for the imagery used. Therefore, a radiative transfer model in Atmospheric and Topographic Correction (ATCOR) module in Erdas Imagine 2013 was used for atmospheric correction. ATCOR masks haze, cloud, water and enhances pixel visibility. In this study, we used the MODerate resolution atmospheric TRANsmission (MODTRAN) code to retrieve the atmospheric parameters for ATCOR from the look-up table as ground-based reflectance and atmospheric data were unavailable. Digital number values were then converted to reflectance based on metadata provided with the Landsat images (Chander et al. 2009, Guanter et al. 2009, ERDAS and Geosystems 2011).

Image classification

A supervised maximum likelihood classifier was adopted for classification (Liu et al. 2002, Manandhar et al. 2009, Tseng et al. 2008, Xi 2007). The technique is based on statistical probability that assigns pixel values to the category with the highest likelihood (Aldrich 1997, Dean and Smith 2003, Ince 1987). Spectral signatures were created and applied in categorizing similar pixels in the entire image using eight polygons representing training data sets for each habitat class. A color composite of 3, 4 and 5 bands were used to facilitate visual interpretation while the Gaussian distribution function was applied in the stretching process. The image was classified into four class categories namely: Woodland, Grassland, Dense forest and Less dense forest. A total of 82 field ground data points, archival high resolution aerial photographs, interviews and expert opinion were used to validate the classified images. Confusion matrices were then created to compare reference data with the maximum likelihood prediction and for calculation of the overall accuracy (OA), producer's accuracy (PA) and user's accuracies (UA). Overall accuracy is a percentage (%) between correctly classified classes and the total number of test reference data, while producer's accuracy is the probability of a specific class being correctly classified. User Accuracy is the possibility that a sample of a specific class represents the category on the ground.

Modelling habitat fragmentation

Fragstats metrics were extracted from all processed Landsat images. Fragstat metrics offer a distinct capacity to determine a landscape's spatial configuration, hence valuable in understanding landscape change arising from fragmentation (Cushman 2006, Jorge and Garcia 1997, Saikia et al. 2013, Millington et al. 2003). All classified images were converted to ASCII format in ArcGIS 10.2. A C-program, a raster version inbuilt within Fragstas that accepts ASCII image files was applied using the eight cell rule. The ASCII format scenes were imported into Fragstats and ASCII built-in-algorithm se-

Table 1. Fragmentation Indices used in the current study.

Fragstats Metrics	Description
Patch Density (PD)	Number of patches of the corresponding patch type.
Largest Patch Index (LPI)	An index used to quantify the percentage of total landscape area characterized by the largest patch.
Edge density (ED)	Used to assess edge length per unit area.
Patch Number (NP)	A measure of the magnitude of fragmentation of patches
Interspersion Juxtaposition Index (IJI)	A measure of adjacency of patches determined by dividing the length between patch edge by the number of patches within a landscape. Values approaching 0% indicate that a patch is adjacent to only one other patch and 100% indicate that a patch is in similar proximity to multiple patches within a landscape.
Patch Area (MN)	The sum across all patches in the landscape of the corresponding patch metric values, divided by the total number of patches. Expressed in hectares.
Perimeter Area Ratio-PARA	Refers to the ratio of the patch perimeter (m) to area (m^2).
Total Area (CA)	Refers to the sum of areas (m^2) of all patches for the patch type.
Percentage of Landscape (PLAND)	Useful in computing the proportional abundance for each of the patch type across the landscape.

lected for running the Fragstats model. Three multi-level structure metrics were selected at patch, class and landscape level (McGarigal and Cushman 2002). Metrics relevant in explaining the magnitude and extent of fragmentation were then selected from the 1975, 1995 and 2012 image scenes. A total of 155 samples were randomly selected and extracted. As recommended by McGarigal and Cushman (2002) two metrics i.e. perimeter area relationship and patch area were statistically used to test the magnitude of fragmentation. Mann-Whitney U and Post hoc ANOVA tests were used to evaluate differences among patch areas in all the years. The Games-Howell was used to determine forest fragmentation. The indices used in this study are briefly described in Table 1.

Results

Classification and accuracy assessment

The overall accuracy for 1975, 1995 and 2012 image scenes was 78.26%, 84% and 76.54% respectively (Table 2). Changes in total area coverage were observed in all years (Figures 2a, b and c).

Change detection

The study findings showed substantial land modification in most of the cover types during the study period i.e. decline in dense forest (31, 675.70 hectares) and less dense forest (by

Table 2. Accuracy assessment tests (Producer's Accuracy - PA), User's Accuracy - UA).

Habitat Class	1975		1995		2012	
	PA (%)	UA (%)	PA (%)	UA (%)	PA (%)	UA (%)
Dense Forest	100	75	100	100	80.77%	95.45%
Less Dense Forest	66.67	100	100	100	100.00%	60.00%
Woodland	66.67	100	66.67	66.67	75.00%	58.54%
Grassland	100	100	100	100	84.62%	91.67%
Overall Accuracy	**78.26**		**84**		**76.54%**	
Kappa co-efficient	**0.7416**		**0.812**		**0.7284**	

Figure 2. Land use land cover (LULC) maps in 1975 (**a**), 1995 (**b**) and 2012(**c**).

11, 267.38 hectares) and increase in grassland (21, 230.01 hectares). However, changes in areas covered by woodland were inconsistent, i.e. increase by 15,884.46 hectares between 1975 and 1985 and decline by 8, 182.03 between 1985 and 2012) – Figure 2.

Fragmentation trends

Temporal variability in fragmentation

Dynamic fragmentation trends were observed (Table 3). Patch number was relatively higher in dense forest and woodland in 1975, 1995 and 2012 than in less dense forest and grassland. The highest percentage of landscape (PLAND) were recorded in less dense forest than the rest of the habitats while woodland and less dense forest habitats had the highest edge density (Figure 3a–c). Furthermore, dense forest showed the most declining patch number during the study period. An analysis of the largest patch index (LPI) showed that less dense forest had the highest LPI, while woodland, dense forest and grassland had the least values, below five. Woodland had the highest PARA compared to the rest of the habitat types (Table 3).

Spatial variation in fragmentation

Study findings indicated a higher probability of dispersion linked to woodland and less dense forest. Interspersion Juxtaposition Index (IJI) ranged between 0 (for clumped patches) and 100 (for grassland). In 1975 and 1995, the grassland habitat had the highest IJI while in 2012, less dense forest had the highest IJI. The interspersion juxta-

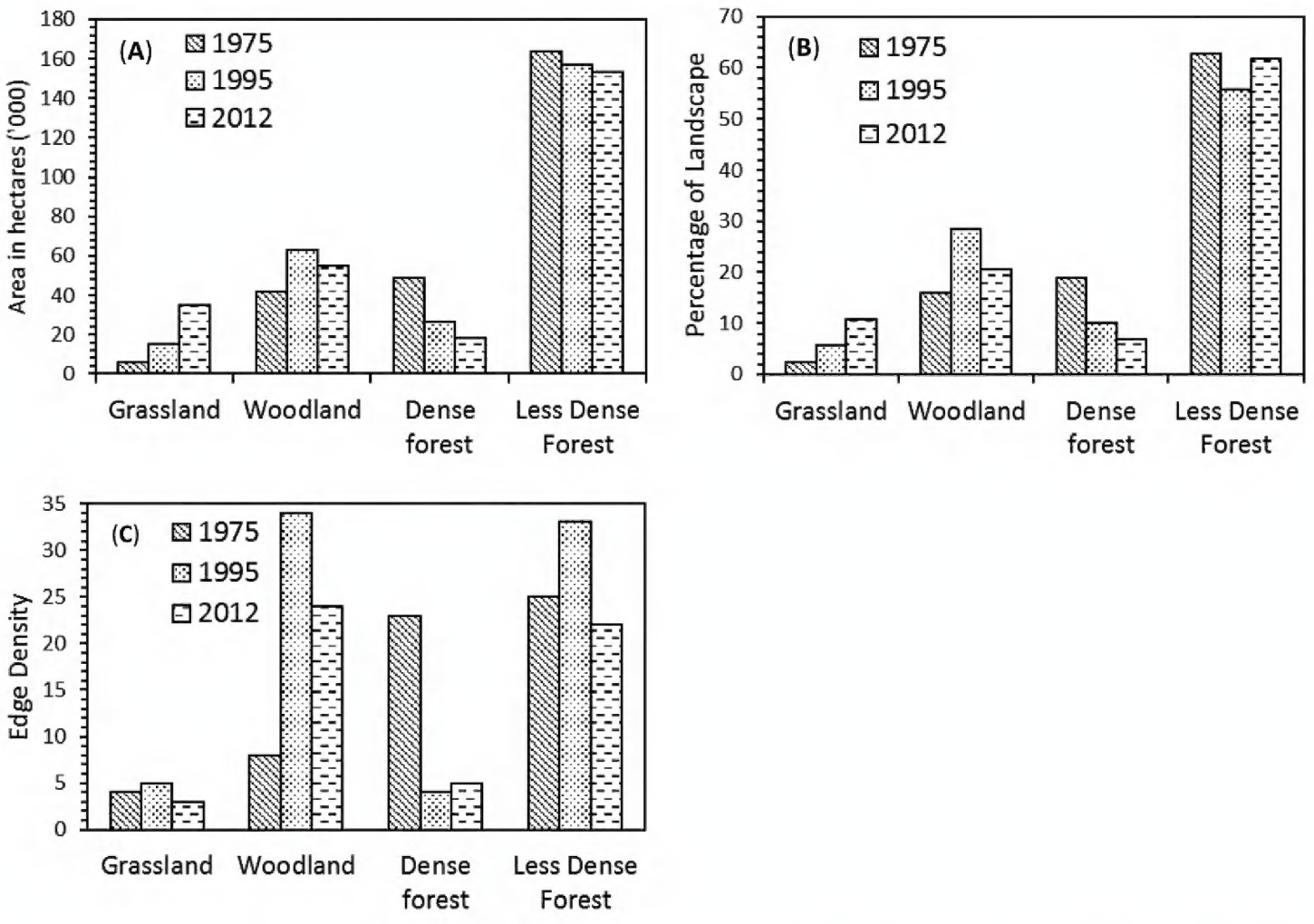

Figure 3. Temporal patterns of total area coverage (**A**), percentage of landscape (**B**) and edge density (**C**).

Table 3. Patch area compared by Mann-Whitney Tests.

Class	Year	z-value (1975–1995)	*Prob* >|z|	*z-value* (1995–2012)	*Prob*>|z|
Dense forest	1975 1995 2012	9.495***	0	-6.872	0.1895
Grassland	1975 1995 2012	13.680***	0	-7.441***	0
Less dense forest	1975 1995 2012	16.728***	0	-8.268***	0
Woodland	1975 1995 2012	-16.63***	0	2.461***	0

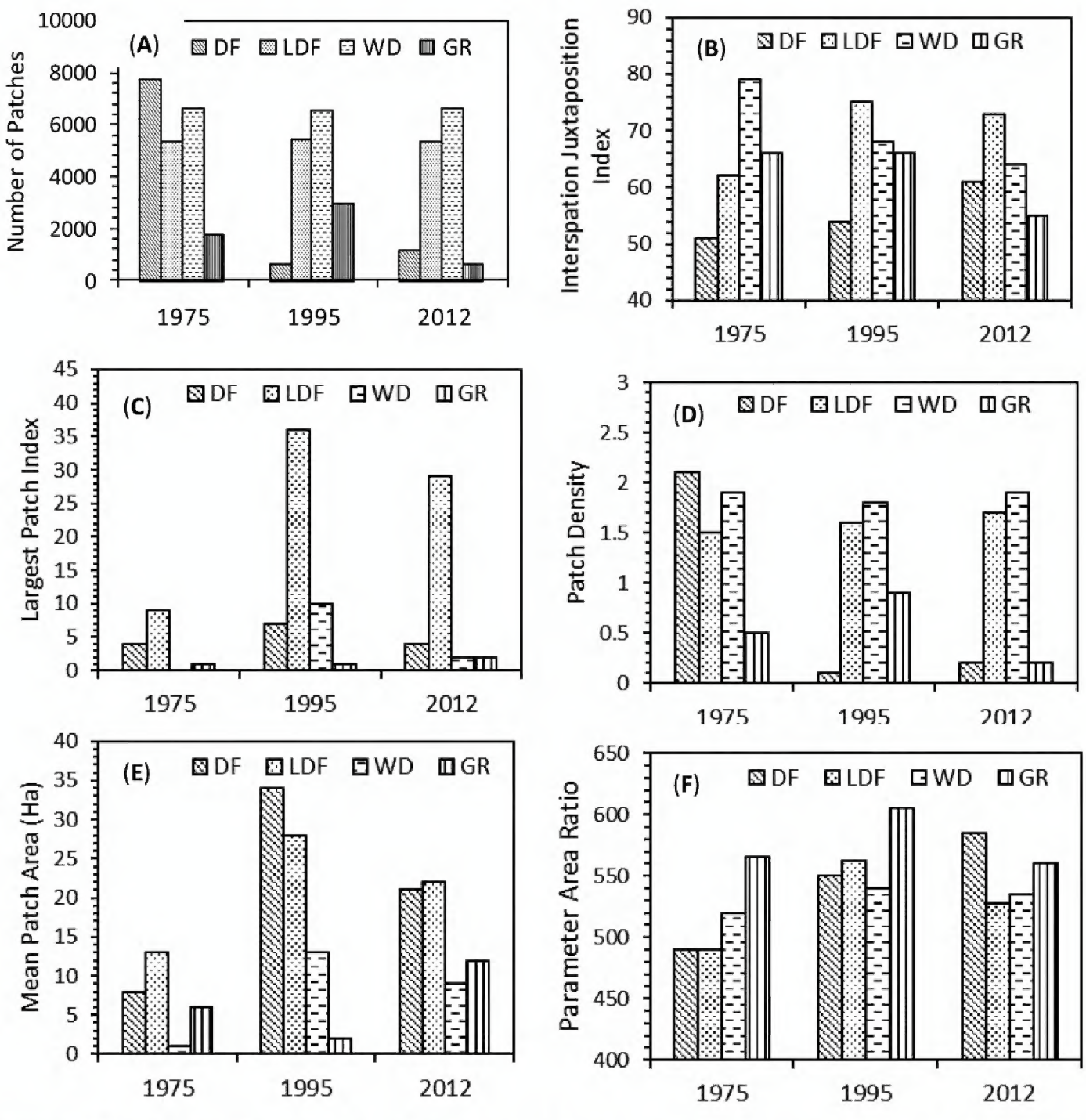

Figure 4. Spatial variability in number of patches (**A**), Interspersion Juxtaposition Index (**B**), Largest patch index (**C**), Patch density (**D**), Mean patch area (**E**) and Perimeter area ratio (**F**) in 1975, 1995, and 2012. DF- dense forest, LDF-less dense forest, WD-woodland, GR-grassland.

position index (IJI), was useful in characterizing the degree of adjacency for each patch type e.g. Burgess et al. (2007). Additionally, the largest patch number and mean patch area was evident in dense forest in 1975 and woodland in 2012 (Figure 4).

Mann-Whitney test results

Mann-Whitney tests were applied to the data. Mann-Whitney test results showed distinct differences in patch area ($p<0.01$) as summarized in (Table 3). These results were strong indicator of a rapidly fragmenting landscape.

Games Howell test results for perimeter area relationship

Game-Howell test is ideal for unequal sample sizes charaterised by heterogeneity and has been widely used in vegetation mapping that include taxonomic profiles in the Atlantic and Caatinga biomes of northestern Brazil (Pacchioni et al. 2014), forest transformation in Uluguru mountains (Ojoyi et al. 2015), the effect of fire on Penderosa pine forest density, canopy cover, tree size and basal area (Stephens et al. 2015) and shrub density in Zegros forest, southwest Iran (Askari et al. 2013). Games Howell test results showed significant patterns of fragmentation between 1975 and 1995 in all habitats ($p \leq 0.05$). In 1975 and 2012, the trend was significant in less dense forest and woodland ($p \leq 0.05$), while in 1995 and 2012, the trend was significant in grassland, dense forest and less dense forest ($p \leq 0.05$) (Table 4). A highly significant trend with perimeter area relationship was evident with less dense forest across the years.

Discussion

This study showed a progressive fragmentation at both spatial and temporal domains. Variability in responses to fragmentation was also noted for different habitats. Fragmentation in the area is not only dependent on topography but also adjacency to land for agriculture, urbanization/settlement and infrastructure development, which

Table 4. Games-Howell tests for the mean parameter area ratio (PARA) in 1975, 1995, 2012.

Class		Mean			*p* value	
	1975	1995	2012	1975 vs 1995	1975 vs 2012	1995 vs 2012
Grassland	565.28	606.21	560.00	0.0001	0.596	0.0001
Dense forest	498.12	549.14	483.7	0.0001	0.3	0.0001
Less dense forest	496.29	563.06	529.5	0.0001	0.0001	0.0001
Woodland	498.58	535.43	534.3	0.0001	0.0001	0.893

are considered key drivers of landscape transformation in the region. All these anthropogenic activities contribute to habitat losses and species decline. Implications on the landscape are presented with a reflection on policy and future management.

Habitat modification

There was a transformation in habitat extents within the study area. Significant losses were recorded for dense forest (31, 675.70 hectares) and less dense forest (by 11, 267.38 hectares), however, there was a steady increase in areas covered by grassland. Based on field study observations, these changes can be attributed to expanding agricultural fields and increased exploitation of timber and non-timber products to meet the increasing urbanization demand in Morogoro district. This finding is in agreement with Burgess et al. (2001) and Burgess et al. (2002) who found a substantial decline of dense forest in the Uluguru mountains due to urbanization and agricultural (Burgess et al. 2002, Burgess et al. 2001). In other parts of Tanzania, related studies established effects of reduced tree density to land modification (Yanda and Shishira 1999, Munishi et al. 2010). Habitat modification could also be attributed to general population increase in non-urbanized areas, also known to influence its spatial configuration (Fischer and Lindenmayer 2007).

Spatial and temporal variation

As aforementioned, there was a general decrease in area covered by dense and less dense forest habitat. A decreasing trend in the extent of total habitat coverage relates to deleterious fragmentation as effects of habitat fragmentation are dependent on habitat size (Fahrig 2003). Furthermore, perimeter-area results in this study show distinct differences in woodland and grassland habitats. In most instances, high perimeter-area relationship characterizes rapid rate of fragmentation underlying the two landforms e.g. Jha et al. (2005) and McGarigal (2006). Woodland habitat displays a patchy type of deforestation, shown by an increased patch number between 1975 and 2012. The slight decline in patch number can be attributed to the strong traditional leadership forest maintance authority in the 1970s, a responsibility that has now been taken over by the Tanzanian Government that permits logging and farm allocations. Dynamics in mean patch area were observed in the woodland and less dense forest. Notable was the gradual decrease in patch size, while patch number increased by 412 and 391 in dense forest and woodland respectively, an indication of fragmentation patterns in the area earlier observed by Jha et al. (2005). On the other hand, patch area was ideal in characterizing distinct areas with analogous environmental conditions, where patch boundaries are distinguished by discontinuities in environmental character states relevant to the organism or ecological phenomenon under consideration. A combination of patch density (PD), PARA and mean nearest neighbor distance are considered

profound in estimation of the extent of fragmentation in each of the habitats analyzed (Jha et al. 2005). Patch density and PARA are regarded as important in fragmentation assessments, particularly in natural ecosystems because they have a strong influence on ecosystem functioning and ecological processes (McGarigal 2006).

Similarly, a distinct variation in patch number was observed. Woodland and less dense forest had the highest patch number across the years. This can be attributed to the great extent of fragmentation resulting from natural resource exploitation. Furthermore, their vicinity to Morogoro town and management by local authorities may be possible drivers increasing their susceptibility to fragmentation (Fahrig 2001, Fahrig 2003, Wiens 1995, McGarigal 2006, Fischer and Lindenmayer 2007). The woodland habitat had a relatively greater patch density, signifying higher spatial heterogeneity. In addition, the largest patch index was associated with less dense forest while least values were associated with the grassland habitat. This provided information on least and most fragmented landscapes, a good indicator of minimum area requirements for species survival (McGarigal 2006). In addition, the largest patch index, another good indicator for species survival was significant in the less dense forest compared to the rest of the habitats (Rutledge 2003).

Dense forest and woodland had the greater edge density. This could be attributed to increased exposure to farmlands and settlements prevalent in the area. Edge effects characterize the biophysical state of ecosystems at the periphery or in the neighborhood. This is because increased habitat fragmentation exposes habitat to edge effects, compromising the ability of an ecosystem to provide relevant goods and services (Murcia 1995). This limits a habitat's long-term ability to sustain a population as it intensifies species mortality rate (Fahrig 2003). It also influences occurrence of native species populations (Murcia 1995) and ensures that the interaction of species in disturbed environments remains restricted, advancing their mortality risk (Rutledge 2003). Related literature also found a high intensity of fragmentation associated with more edge effects through exposure of contiguous habitats to solar radiation and soil moisture to drier heat conditions (Rutledge 2003).

Games-Howell test results showed a significant level in the perimeter area relationship ($p \leq 0.05$). This could be explained by the fact that less dense forest adjoins dense forest, taking up regions dominated by woodland. It is also possible that the on-going fragmentation is a major driver of conversion of dense forest and woodland to less dense forest. Potential socio-economic drivers could be a result of the expanding Morogoro town and increasing agricultural fields in the adjacent local regions. Similarly, other studies showed how adjoining activities influence intact habitat ecosystems as a result of their structural configuration (Echeverría et al. 2007).

Drivers to habitat fragmentation and conservation implications

Anthropogenic activities significantly influence habitat fragmentation in the region. For instance, extensive farming and urban growth are possible drivers to habitat modi-

Figure 5. Drivers to fragmentation, note the settlements in the valley and cleared forest in the background and fore ground for crop farming and grazing, respectively (**A**) and small scale maize and banana fields within the forest in (**B**).

fication and fragmentation. The area has a conducive montane climate that supports subsistence farming, a prevalent socio-economic activity in the region (Burgess et al. 2007, Swetnam et al. 2011, Yanda and Shishira 1999). This seems to significantly influence all the four habitats. Increasing population growth and consequent increase in settlement and farmlands may have extirpated important fauna and flora in the Ulugurus (Bjørndalen 1992, Burgess et al. 2002, Burgess et al. 2001, Burgess et al. 2007, Swetnam et al. 2011, Hall 2009, Yanda and Shishira 1999) – Figure 5. Habitat fragmentation in the study area can also be attributed to a complex nexus of socio-economic processes (Kessy et al. 2016, Rosales 2008). These processes act at various scales i.e. international (global forest products market growth, commercialization and urbanization), national (changing population, growing local markets and national legislation and governance) and local conditions (livelihoods and levels of poverty) (Wehkamp et al. 2015, FAO 2007, Daly and Farley 2004, Czech 2013). Kessy et al. (2016) for instance notes that local and international demand for timber and agricultural commodities in a globalizing world are major drivers to forest fragmentation in the area. Globalization, with its characteristic scramble by the developing countries to increase their market share on the global marketplace has increased pressure on existing forests and forest land (Hecht and Saatchi 2007, Rosales 2008).

To forestall some of the problems earlier highlighted, the study area, identified as biodiversity hotspots with important ecological functions such as groundwater recharge, surface flow and animal habitat need to be protected from the impacts of land modification and fragmentation. Implications of habitat modification and fragmentation in Morogoro region can be better deciphered through the impact on habitat structure and species losses. The increased habitat losses, mainly attributed to anthropogenic factors may negatively influence genetic diversity and lead to losses of potentially useful genes originally accommodated in intact areas (Ojoyi et al. 2015, Burgess et al. 2007, Swetnam et al. 2011, Hall 2009, Yanda and Shishira 1999, Shirima et al. 2011). Therefore, we recommend that mitigation measures should be adopted to ensure pro-

tection and management of these fragmenting habitat ecosystems. To optimize mitigation measures, the adverse effects of habitat modification and fragmentation need to be understood by all stakeholders. In addition, policy measures and sustainable bottom-up approaches to management and conservation of forest resources should be instituted in the region.

Conclusions

Distinct differences in magnitude of fragmentation were evident across the four habitat categories. The study findings show that fragmentation was highest in less dense forest. Subsistence farming, increasing human population and urban growth are thought to be key drivers to habitat modification and fragmentation, hence it is concluded that anthropogenic processes are the major drivers to habitat fragmentation in the area. The fragmenting landscape is expected to significantly influence floral and faunal vulnerability, likely to compromise the area's ability to among others assimilate organic carbon and to supply socio-economic and environmental goods and services. It is therefore necessary that the study area, and indeed the entire eastern arc mountains region be protected from the impacts of land modification and fragmentation. The study further underscores the value of satellite imagery in concert with relevant reference data in understanding spatio-temporal transformation of vulnerable landscapes arising from anthropogenic processes.

Acknowledgements

The current study was financed by UNESCO L'Oreal Foundation for Women in Science, The International Development Research Centre (IDRC), Ontario Canada and University of KwaZulu-Natal Post-Graduate sponsorship. Training by the University of KwaZulu-Natal, South Africa and Faculty of Geo-Information Science and Earth Observation, University of Twente, Netherlands is greatly appreciated. Field support provided by Sokoine University of Agriculture, forestry department, Tanzania Government Forestry departments in Morogoro region and the Wami/Ruvu Basin Office is highly acknowledged.

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